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LCA FOR RENEWABLE RESOURCES

Potential for optimized production and use of rapeseed biodiesel. Based on a comprehensive real-time LCA case study in Denmark with multiple pathways

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Abstract
Purpose Several factors contribute to the current increased focus on alternative fuels such as biodiesel, including an increasing awareness of the environmental impact of petrochemical (PC) oil products such as PC diesel, the continuously increasing price of PC oil, and the depletion of PC oil. For these reasons, the European Union has enacted a directive requiring each member state to ensure that the share of energy from renewable sources in transport be at least 10 % of the final consumption of energy by 2020 (The European Parliament and the Council 2009). This LCA study assesses the specific environmental impacts from the production and use of biodiesel as it is today (real-time), based on rapeseed oil and different types of alcohols, and using technologies that are currently available or will be available shortly. Different options are evaluated for the environmental improvement of production methods. The modeling of the LCA is based on a specific Danish biodiesel production facility.
Methods The functional unit is "1,000 km transportation for a standard passenger car." All relevant process stages are included, such as rapeseed production including carbon sequestration and N_2O balances, and transportation of products used in the life cycle of biodiesel. System expansion has been used to handle allocation issues.
Results and discussion The climate change potential from the production and use of biodiesel today is 57 kg CO_2-eq/1,000 km, while PC diesel is 214 kg CO_2-eq/1,000 km. Options for improvement include the increased use of residual straw from rapeseed fields for combustion in a power plant where carbon sequestration is considered, and a change in transesterification from a conventional process to an enzymatic process when using bioethanol instead of PC methanol. This research also evaluates results for land use, respiratory inorganics potential, human toxicity (carc) potential, ecotoxicity (freshwater) potential, and aquatic eutrophication (N) potential. Different sources for uncertainty are evaluated, and the largest drivers for uncertainty are the assumptions embedded into the substitution effects. The results presented should not be interpreted as a blueprint for the *increased* production of biodiesel but rather as a benchmarking point for the present, actual impact in a well-to-wheels perspective of biodiesel, with options for improving production and use.
Conclusions Based on this analysis, we recommend investigating additional options and incentives regarding the increased use of rape straw, particularly considering the carbon sequestration issues (from the perspective of potential climate change) of using bioalcohol instead of PC alcohol for the transesterification process.

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1 Introduction

The European Union has enacted a directive requiring each member state to ensure that the share of energy from renewable sources in transport be at least 10 % of the final consumption of energy by 2020 (The European Parliament and the Council 2009), and the total energy consumption for transport in 2020 is currently projected to be 438.6 million tons of oil equivalent (European Commission—Directorate-General for Energy and Transport 2008). The production of biodiesel in Europe in 2008 was 5.5 million tons (Emerging Markets Online 2011). As such, the demand for energy from renewable sources is fixed, and the main concern that remains is *how to reach this target with the lowest possible environmental impact.*

This paper is based on an LCA research program for biodiesel that has operated for 3 years, 2009–2012. Two Danish companies—Emmelev A/S and Novozymes A/S—have been partners in the program, with a focus on optimization of the environmental performance of biodiesel in a well-to-wheels (WTW) perspective.

To handle this optimization problem, we apply terminology from the optimization methodology as outlined in Montgomery (2005), and as an inspiration in this LCA study. In optimization methodology, explanatory variables are used to track changes in a model and enable a structured and transparent analysis of such changes, and to determine how (and how much) these changes lead to changes in the response variables/impact categories for the system under investigation. We take the same approach in this paper. Different explanatory variables—including transesterification process, type of alcohol, and use of biomass raw material—have been identified regarding the production and use of biodiesel in order to potentially give a better or worse response for the environmental impact categories. Other explanatory variables are presented in the Electronic Supplementary Material.

The initial project focuses on the transesterification process, in which either an enzymatic or conventional transesterification can be applied. The other explanatory variables are used for benchmarking the potential of the transesterification process with this explanatory variable.

Harding et al. (2008) develop an LCA of biodiesel production and compare enzymatic and conventional transesterification processes in a well-to-tank perspective with multiple impact categories, finding that enzymatic biodiesel transesterification is environmentally advantageous compared to conventional biodiesel transesterification. Malça and Freire (2011) present a comprehensive review of 28 different LCA studies on biodiesel in Europe, in which all results are evaluated based on greenhouse gas (GHG) emissions per MJ. The two main issues raised in this study are the variability of results and the different modeling approaches between the different LCAs. The different modeling approaches are explained by different assumptions regarding geographical scope, the functional unit, multifunctionality (i.e., allocation problems), and agricultural modeling (primarily N_2O emissions). Other modeling differences are also mentioned, which we regard as "prospective"—i.e., answering the questions of what can happen, as opposed to studies of "the current situation" that are based on observable processes. These GHG emissions are reported to range from 15 to 170 kg CO_2-eq/GJ. According to Howarth and Bringezu (2009), some, but few, biofuel studies report on environmental impacts other than GHGs.

Here, our study addresses multiple environmental impacts—including toxicity modeling based on the USEtox^{TM} methodology, nutrient balance calculations in the agricultural stage, and land use—and discusses the indirect land use change (ILUC) impacts. Production data is based on empirical data from a Danish biodiesel producer. Current production technology has been modeled with state-of-the-art LCA methodology, and can be considered a benchmarking point for improving already-established biodiesel production and use in Denmark. Furthermore, options that may reduce environmental impacts are investigated for the processes used in different biodiesel production variables.

2 Materials and methods

2.1 Goal and scope definition

The goal of this study is to present a full comparative and quantitative LCA of biodiesel from rapeseed oil in Denmark. Two baselines for benchmarking are developed: (1) the environmental impact of the production and use of biodiesel based on rapeseed oil (named PWB1), and (2) the environmental impact from the production and use of PC diesel (named PWD0). Different pathways with alternative production technologies are developed to investigate potential environmental improvements for biodiesel production and use. These results will allow better and more-informed support for decisions regarding the future production and use of biodiesel.

This LCA study addresses decision-makers involved in the direct production of biodiesel, and decision-makers involved in developing policies for biofuels.

The LCA is based on current technologies or near-future technologies. The functional unit for our system is 1,000 km driven by a passenger diesel car with a 20 % blend of biodiesel (B20). The European governmental directive (The European Parliament and the Council 2009) aims for a 10 % ratio of biofuel used in transport in member countries by 2020. However, due to project requirements our analysis focuses on a 20 % blend. The passenger diesel car is based on an EcoInvent process (Operation, passenger car, diesel, fleet average 2010/RER U), which reflects a fleet average in Europe in 2010. The study includes tailpipe emissions, biodiesel production, oil production, alcohol production, and rapeseed production—including specific modeling of fertilizer and pesticide emissions. Our study assumes that biogenic CO_2 emission into the atmosphere is balanced by an equal uptake of carbon by growing new crops in the production system (in the next time-period). Hence, all biogenic CO_2 emission is accounted with zero impact, while CO_2 emission from PC diesel is accounted as increased. The product system is illustrated in Fig. 1.

The choices modeled here are based on Fig. 1. For the transesterification variable, the choice is between (1) enzymatic catalyzed transesterification, in the following termed "Enzymatic 1," (2) different enzymatic catalyzed transesterification, in the following termed "Enzymatic 2," and (3) a conventional transesterification process using sodium hydroxide catalyst. For the alcohol used in the production variable, the choice is between (1) PC methanol, (2) PC ethanol, and (3) bioethanol. For the biomass raw material variable, the choice is (1) not to use straw for power generation in a power plant (I), (2) to continue using the same amount of residual straw as at present for power generation in a power plant (II), and (3) using an increased amount of rape straw compared to the present amount (III). For the transport variable, the choice is between (1) short transport distance (local) and (2) long transport distance (regional). The long transport distance in the transport variable cannot be considered consistent with inventory data from Denmark, but it nonetheless seems relevant to include it for benchmark options. Different combinations of these options are considered as different PWs; at present not all of these combinations are technically possible or economically feasible, but they might still be interesting for further policy development or research.

Based on these choices, there are 54 possible combinations. Two baselines (PWD0 and PWB1) and seven of the 54 alternative pathways (PWB2 to 8) for comparison with the baselines are presented in this paper. These pathways are outlined in Table 1, which includes the following abbreviations: *D* = PC diesel, *B* = rapeseed biodiesel, and each ID number identifies the unique combination.

The data for this LCA was collected between 2009 and 2011. Based on Makridakis et al. (1998), the modeling of PWD0 and PWB1 conducted in this paper addresses the time period t_P = present (~2010). Data for PWD0 and PWB1 reflects average production data in Denmark as it is today (real-time) .[1] Hence, no assumptions are made about what will happen in the future for this data, or about what will happen if production is *increased*. Forecasting of PWD0 and PWB1 is done by the "naïve forecast method" (Makridakis et al. 1998), which assumes that the best forecast for the future is the current value of the time series given the information available during our research. This also implies that our study is not strictly comparable to the study by Edwards et al. (2008), which addresses potential environmental impacts from only increased production.

System expansion was used to solve allocation issues. This allocation method is the preferred method for solving allocation issues according to International Standard Organization (ISO) norms (ISO 2006a, b). System expansion has been based on literature surveys and specialist knowledge, and product substitution was modeled the way it is believed to be currently.

As a point of departure, EDIP 2003 methodology (Hauschild and Potting 2005) was chosen as the primary impact assessment methodology. However, not all of the presented impact categories were available in EDIP 2003. As such, other methodologies have also been used, based primarily on the criterion that they should be the newest available. Environmental impact is evaluated based on the following six impact categories:

1. Climate change potential based on EDIP 2003 (Hauschild and Potting 2005)
2. Land use based on Recipe (Goedkoop et al. 2008) and Impact 2002+ (Jolliet et al. 2003)
3. Respiratory inorganics based on Humbert et al. (2011)
4. Human toxicity (carc) based on USEtox™ (Rosenbaum et al. 2008)
5. Ecotoxicity freshwater based on USEtox™ (Rosenbaum et al. 2008)
6. Aquatic eutrophication (N) based on EDIP 2003 (Hauschild and Potting 2005)

This study is based on data from a biodiesel producer in Denmark, and from technical research at the Chemical Engineering Department of The Technical University of Denmark. Other data has been found in the literature, and remaining data is from the EcoInvent database. Assumptions and data are cited

[1] When data for the present time period was limited, assumptions were made to fit data to this criterion.

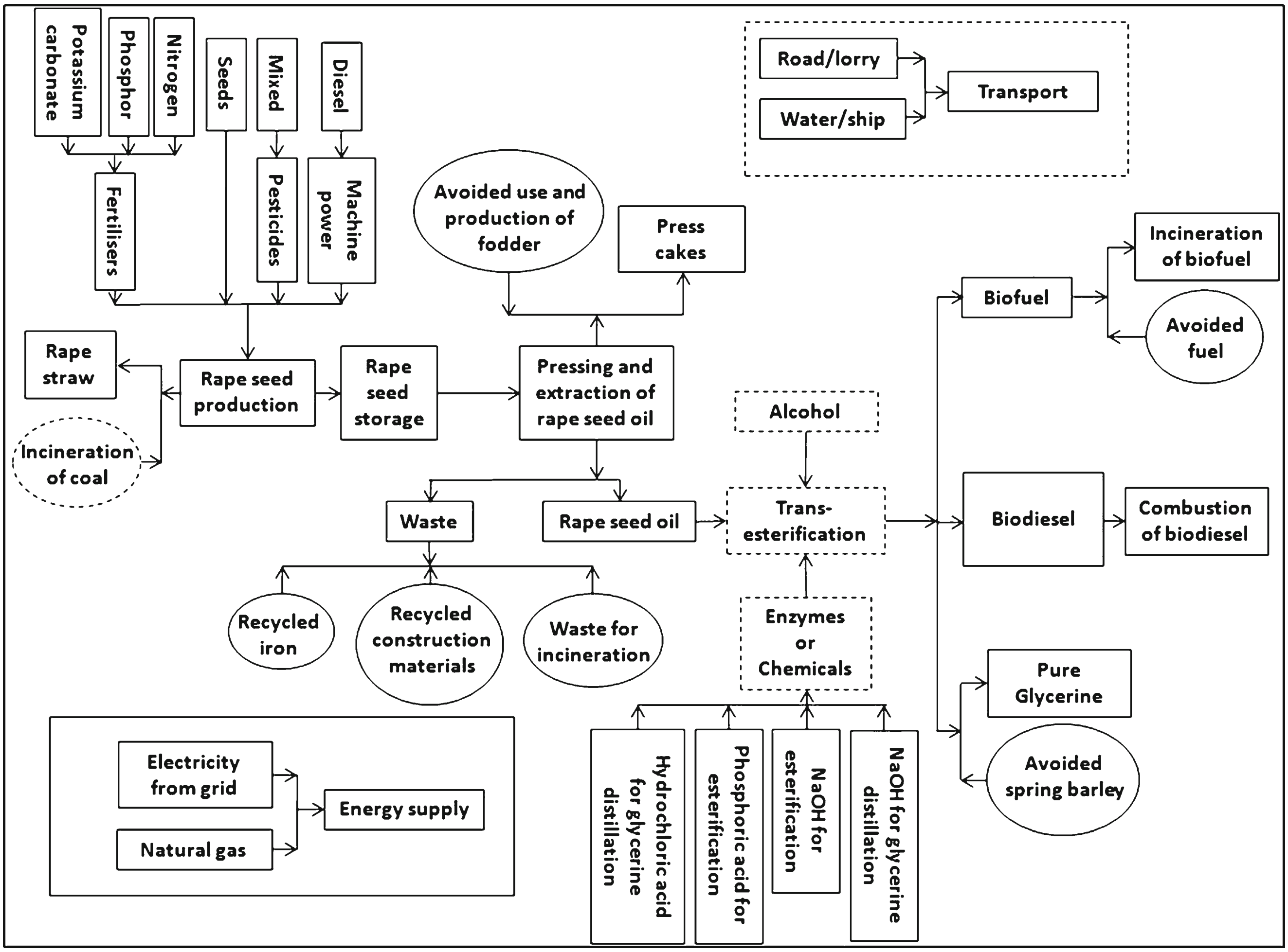

Fig. 1 The analyzed system for production and combustion of biodiesel for passenger car transport based on rapeseed oil. Energy supplied for the pressing and oil extraction process and transesterification process is average Danish grid mix together with natural gas. Transportation includes road and water transport, mainly for the transport of seed to the pressing and extraction process. The *dashed lines* illustrate the variables that will or can be changed to create alternative pathways (PWB2 to 8)—see Table 1

in the following lifecycle inventory subsections; however, some of the data applied is classified, and cannot be published due to the project stakeholder's business opportunities. The environmental modeling tool SimaPro (version 7.2) has been used, including the EcoInvent database version 2.0.

2.2 Lifecycle inventories

2.2.1 Rapeseed production

The unit process "rapeseed production" models the emissions from rapeseed production under average Danish conditions, scaled to 1 ha×year. When modeling fertilizer use in Denmark, it is relevant to consider two different types of rapeseed fields—those near plant farms, and those near pig farms. The emissions of different nutrients from a 1 ha×year agricultural oilseed rape field are calculated using nutrient balances of plant farms and pig farms and two different soil types (sandy loam and coarse sandy soil, according to Danish standards). Subsequently, the emissions have been averaged across the different groups by the relative frequency of plant farms or pig farms, on sandy loam soil or coarse sandy soil, in Denmark (data is presented in Table S1 in the Electronic Supplementary Material).

Nitrogen balances For each of the four combinations of soil type and farm type, a nitrogen balance has been calculated. In the Electronic Supplementary Material, Fig. S1 is presented to illustrate the nitrogen balance. There are two major inputs of nitrogen to the field: one from nitrogen fertilizers, and one from atmospheric deposition. The major outputs are removal of seeds and straw, ammonia volatilization, denitrification in the form of N_2O and N_2, and nitrate leaching.

The fertilizer input of nitrogen is based on the nitrogen norms defined for the specific soil type in Danish regulations

Table 1 Different pathways for biodiesel production and use discussed in this paper

Name	Biodiesel production variables			
	Alcohol production	Transesterification	Biomass raw material	Transport
PWD0	No	No	I	No
PWB1	PC methanol	Conventional	II	Short
PWB2	Bioethanol	Conventional	II	Short
PWB3	Bioethanol	Enzymatic 1	II	Short
PWB4	PC methanol	Enzymatic 2	II	Short
PWB5	PC ethanol	Conventional	II	Short
PWB6	PC methanol	Conventional	III	Short
PWB7	PC methanol	Conventional	I	Short
PWB8	PC methanol	Conventional	I	Long

PWD0 and PWB1 are both considered as real baselines because they represent current actual production and use; each ID number is used to identify the unique combination

PW pathways, *D* PC diesel, *B* rapeseed, *I* 0 t/(ha×year), *II* 0.52 t/(ha×year), *III* 0.86 t/(ha×year)

(Naturerhvervsstyrelsen 2010). In these regulations, the "mineral fertilizer equivalencies" (MFE) for different types of fertilizer are also defined—for mineral fertilizer this is 100 %, and for pig slurry it is 75 %. Based on the norm and the MFE, the amount of N added as fertilizer can be calculated.

Fertilizer input is assumed to be in accordance with Danish fertilizer norms (Naturerhvervsstyrelsen 2010) for oilseed rape, i.e., 119 kgNha^{-1} on the coarse sandy soil, and 183 kg Nha^{-1} on the sandy loam. The plant farms are assumed to be fertilized with mineral fertilizer exclusively, while the pig farms are assumed to be fertilized with pig slurry and mineral nitrogen fertilizer. A mineral fertilizer equivalency efficiency of 75 % is assumed for pig slurry. Crop N uptake is estimated as the normative values (Naturerhvervsstyrelsen 2010). Emissions of N_2O and N_2 from fertilizer use are based on the "SIMDEN" model (Vinther and Hansen 2004). Nitrate leaching is calculated from the nitrogen balance. The pig slurry is assumed to have zero climate change impact, since it is considered to be a waste product from pig production. A detailed description of nitrogen balance assumptions/equations is available in the Electronic Supplementary Material.

Phosphorous The amount of mineral P applied on the plant farms is assumed to follow Danish recommendations (Haastrup 2010). On the pig farms, P supply from the animal slurry is assumed to be enough to satisfy the needs of the crops, and therefore no additional mineral P is needed. The amount of P supplied is calculated from the amount of N supplied in the slurry, assuming a P-to-N ratio of 0.11 (Møller et al. 2001). The loss of P varies considerably depending on soil and fertilizer type and climate, but we have no data to quantify this. Instead, a gross average of losses to surface waters estimated to be 0.15 kg×ha^{-1} by Munkholm and Sibbesen (1997) has been used, and other losses are assumed to be minimal. The rest of the P is assumed to be either removed with the crop or accumulated in the field.

Potassium The amount of mineral K applied on the plant farms is assumed to follow the Danish recommendations (Haastrup 2010). On the pig farms, ample K is assumed to be supplied from the animal slurry, and so no mineral K is necessary. The losses of K have not been estimated because losses of K are considered to have no impact on the environment.

Use of pesticides The Electronic Supplementary Material presents the pesticides permitted for use in rapeseed cultivation in Denmark, and the maximum allowed amount of each pesticide to be applied according to Danish Agro (2008). We assume that pesticides have been applied to the maximum permissible level. To calculate emissions, the PEST-LCI (Birkved and Hauschild 2006) model has been used.

Use of rape straw Straw is a co-product of the production of rapeseeds used for biodiesel, and in accordance with the ISO norms system expansion has been applied to solve this allocation issue. Removing straw from the fields for combustion in a power plant (and the associated substitution of coal) can potentially improve the environmental impact of biodiesel. However, removing too much straw from the field leads to a risk of soil organic carbon mining. According to Lafond et al. (2009), it is possible to remove wheat straw from the field without a change in the soil organic carbon if no more than 40 % is removed in 2 out of 3 years (on average 26.7 % p.a.). There will also be no change in the yield of the spring wheat grain, or the grain protein content. It is assumed that these results can be transferred to the production of rapeseed. According to Danish Statistics (2011), the average production of rape straw between 2006 and 2009 was ~3.22 t/(ha×year), resulting in a theoretical possible removal of mass of 0.86 t/(ha×year). However, only 0.66 t/(ha×year) is actually removed according to Danish Statistics (2011), and of this 0.52 t/(ha×year) is used for combustion.

It can be assumed that there is approximately 3 % energy loss in the straw due to a required pre-treatment before it can be co-fired with hard coal in a power plant (Sander 2010, personal communication on *Combustion of straws and substitution of coal at a power plant and ratio of straws used for combustion*, Skærbækværket, DONG Energy A/S,

Denmark). With an energy value of 14.5 GJ/t of straw, the amount of coal that can be substituted is this:

14.5 GJ/(t of straw)×0.52 t/(ha×year)×0.97 (GJ from coal/GJ from straw)~7.3 GJ/(ha×year)

If instead the maximum amount of straw that can be removed (without affecting the long-term carbon content of the field) is used for combustion—i.e., 0.86 t/(ha×year)—then this results in the substitution of coal equivalent to 12.12 GJ/(ha×year).

The emissions from burning rape straw and coal in a power plant are practically equal due to the modern cleaning technology applied in Danish power plants (Jensen et al. 2007). The environmental advantage/improvement results primarily from the emission of biogenic carbon dioxide from wheat straw instead of fossil carbon dioxide from coal.

2.2.2 Pressing and extraction of rapeseed oil

The unit process "pressing and extraction of rapeseed oil" is scaled to 1 t of rapeseed input. According to the LCA Food Database (2006), the production of rapeseed meal is determined by the demand for the rapeseed oil, as the oil is much more valuable than rapeseed meal. The co-production of 1 t of rapeseed meal from rapeseed oil production substitutes soy meal production and barley for animal food with 0.664 and 0.279 t (LCA Food Database 2006). Energy data for pressing and extraction of the rapeseeds is based on Emmelev A/S data.

2.2.3 Biodiesel production

The unit process "production of biodiesel" is scaled to 1 t of rapeseed oil as an input. The major inputs for this process are chemicals (or enzymes), energy, alcohol, and oil. In the case of conventional transesterification process, the input data is based on production information from Emmelev A/S.

Two different processes where enzymes have been used as a catalyst for the transesterification process are modeled: "Enzymatic 1" based on stoichiometry data from Nordblad (2011, personal communication on *Flow data for enzymatic esterification process*, Technical University of Denmark), and "Enzymatic 2" based on Sotoft et al. (2010). The required input of enzymes is based on data from Novozymes A/S. The environmental impacts for 1 kg of enzymes in a cradle-to-gate perspective are based on Nielsen et al. (2007). Both the Enzymatic 1 and Enzymatic 2 processes are based on immobilized enzyme catalysts. Other enzyme processes, including those based on liquid formulated enzymes, could lead to somewhat different results.

Three different types of alcohol have been modeled, (1) PC methanol, (2) PC ethanol, and (3) bioethanol. The alcohols have all been modeled using standard processes from the EcoInvent database. The by-products glycerin and biofuel are presented separately in subsection 2.2.5.

2.2.4 Combustion of biodiesel

The emissions from driving 1,000 km in a diesel passenger car are based on the unit process "Operation, passenger car, diesel, fleet average 2010/RER U," which includes airborne emissions of gaseous substances, particulate matters, and heavy metals. This data has been modified according to data of biodiesel tailpipe emission from the Graboski et al. (2003) report, which is itself based on test data of a "Detroit Diesel Cooperation—Series 60 Engine." Emissions from biodiesel and PC diesel per brake horsepower×hour (bhp-h) delivered at the axle is given in this report. From this data, emissions from 1,000 km delivered have been calculated.

The biodiesel needed for driving 1,000 km is also calculated relative to the EcoInvent PC diesel process. In this unit process, it is estimated that 55.8 kg of PC diesel is consumed per 1,000 km. The calorific energy value for PC diesel is 43.38 GJ/t (International Energy Agency 2005), and for fatty acid methyl ester (FAME) it is 37.362 GJ/t. Test data from Graboski et al. (2003) shows that there is a small decrease in efficiency (for the specific test engine) of the PC diesel and biodiesel, (7.616 MJ/bhp-h)/(7.842 MJ/bhp-h)~3 %. To deliver 1,000 km from the biodiesel, 0.0668 t biodiesel is needed. Table 2 shows the changes in emissions from biodiesel (B20) relative to PC diesel.

The tailpipe CO and CO_2 emissions based on FAME from biomass in Table 2 are considered biogenic, and so these emissions are accounted with zero in this LCA study (as assumed in the Section 2.1).

To account for the contributions of different types of alcohol to the GHG, it is assumed that the average length of fatty acid carbon chains is ~17 C, based on Mattson and Volpenhein (1963). Adding methanol or ethanol to this will increase the length of the carbon chain with 1 or 2 carbon atoms, respectively. Adding bioethanol is accounted for in the tailpipe emission as

Table 2 Relative change in emissions based on Graboski et al. (2003) test data

	THC	NO_x	CO	CO_2	PM	SO_2	VOC
Cert fuel (PC diesel)	1	1	1	1	1	1	1
FAME (rapeseed)	1.05	1.00	0.86	1.00	0.83	0.80	0.82

The changes in emissions are measured per bhp-h delivered at axle. When FAME and FAEE is from biomass, then CO and CO_2 emission is biogenic and can be accounted with zero. It is assumed that the change from FAME to fatty acid ethyl ester (FAEE) results in the same relative change compared to PC diesel

THC total hydrocarbons

being biogenic. The three different ratios applied in this study are then as follows:

- (17 biogenic+2 biogenic)/(19 total) for bioethanol (fatty acid ethyl ester—(FAEE))
- (17 biogenic+2 fossil)/(19 total) for ethanol from fossil resources (FAEE)
- (17 biogenic+1 fossil)/(18 total) for methanol from fossil resources (FAME)

2.2.5 Glycerine and biofuel as by-products

There are two by-products from the biodiesel production process: crude glycerol and impure biodiesel. With the conventional transesterification process, the crude glycerol in PWB1 is purified, which requires the use of chemicals and energy. Two to three percent of the fuel output is considered too impure to meet the specifications to serve as biodiesel; instead, it can be used in an industrial furnace where it is assumed to substitute light or heavy PC fuel oil where the energy value is 2–3 % lower than the pure biodiesel. According to Zijlstra et al. (2009), glycerine can substitute for wheat as feed for pigs. According to Jonasson and Sandén (2004), it can be assumed that the substitution ratio between wheat and glycerine is ~0.93 kg wheat/(kg glycerine). As such, the assumed substitution ratio is a one-to-one energy ratio—that is, 0.93 kg of wheat will be substituted per 1 kg of glycerine produced.

2.2.6 Transportation

Rapeseeds travel from a local farmer to the biodiesel producer by truck over an average distance of 100 km; however, some rapeseeds are from a regional farmer, where transportation is by ship over an average distance of roughly 1,000 km, and then 200 km by truck. These distances are considered relevant for this case, as the Danish producer uses either local domestic rapeseed production or rapeseeds produced in Eastern Europe.

3 Results and discussion

In the following six figures, each impact category is presented for each of the nine PWs. For these results, no emissions from indirect land use change are included—results addressing ILUC are presented separately. Each PW is separated into two parts: tailpipe impact (tailpipe) and production system impact (production). At the top of each graph the aggregated number for both tailpipe and production is presented per 1,000 km. PWD0 and PWB1 represent current production and use, while the rest of the PWs are modeled with changes that are interesting to consider for improved production and use of biodiesel. Furthermore, each PWB in Figs. 2, 3, 4, 5, 6, and 7 illustrates the isolated B20 emission. This is calculated by the following equation: PWB=(B−0.8×PWD0)×5, where B includes both the 80 % PC diesel emission and the 20 % biodiesel emission. Multiplying by 5 gives the 100 % of biodiesel B20 emission. The purpose of illustrating results this way is to highlight the net effects of biodiesel when adding it to PC diesel.

3.1 Climate change potential

PWD0—PC diesel—is the PW with the highest climate change potential, 214 kg CO_2-eq/1,000 km. The tailpipe emission for PWD0 is approximately 180 kg CO_2-eq/1,000 km, while PWB1 has a tailpipe emission of ~12 kg CO_2-eq/1,000 km. The production stage level for PWD0 accounts for ~34 kg CO_2-eq/1,000 km.

The production stage level for PWB1 accounts for ~45 kg CO_2-eq/1,000 km. The change between PWB1 and PWB2 is that, instead of using PC, methanol then bioethanol is modeled. This leads to a decrease in the overall impact by ~9 kg CO_2-eq/1,000 km due to a lower tailpipe emission. However, at present the conventional transesterification process based on ethanol is not technically possible (or economically feasible). The bioethanol is assumed to come from Brazil, and transportation for this is included in PWB2. PWB3 is based on the Enzymatic 1 transesterification process, which makes it possible to use ethanol for the transesterification process. At present, this process seems to be a little less efficient compared to the conventional process in PWB1. It is important to notice here that the conventional transesterification process (PWB1) is a mature technology that has been developed over several decades,

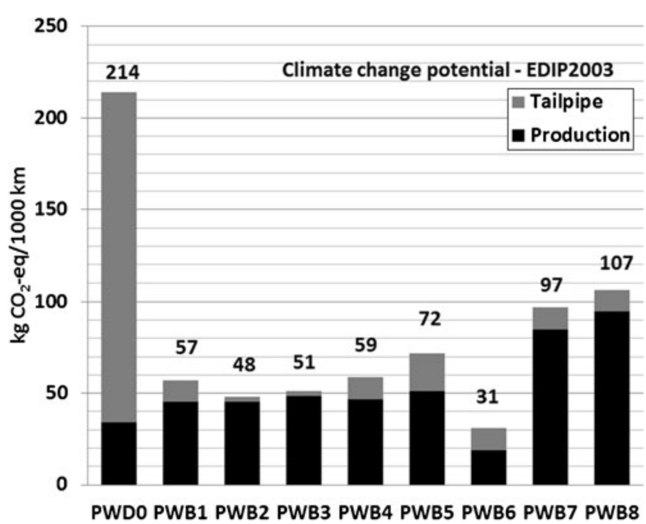

Fig. 2 Climate change potential per 1,000 km driven in a standard diesel passenger car—EDIP 2003

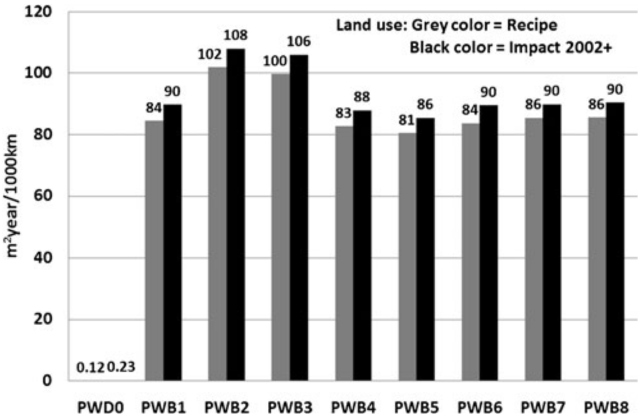

Fig. 3 Land use based on Impact 2002+ and Recipe

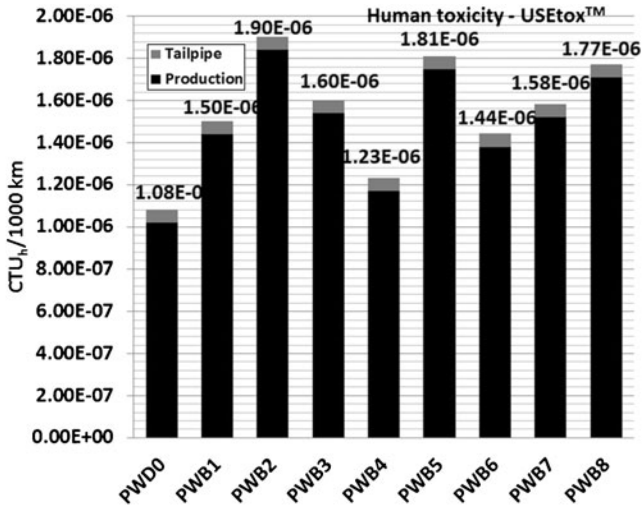

Fig. 5 Human toxicity—USEtox™. *CTU*$_h$ comparative toxic unit, human

while the enzymatic process is a new and rather immature technology. If the enzymatic processes are developed further, it can be expected that there will be a higher potential for improving the technology compared to the already mature and conventional transesterification process. However, no attempt has been done to predict (or forecast) these potentials. This LCA study can serve as benchmarking for further improvement to both technologies. At least two variables can be used for further improvements to the enzymatic process: (1) the mass of enzyme needed per mass of biodiesel out, and (2) the CO_2-eq/t of enzyme produced.

PWB4 is based on Sotoft et al. (2010). PWB4 increases with 2 kg CO_2-eq/1,000 km compared to PWB1, due to a slightly less efficient transesterification process. However, energy data was not transparent from the Sotoft et al. (2010) paper, and so is only roughly estimated here.

PWB6 and 7 are similar to PWB1 except for changes in the amounts of straw used for combustion. PWB6 is modeled with an increased mass of 0.34 t rape straw compared to PWB1, used for combustion in power plants and assumed to substitute for coal resulting in a decrease of 26 kg CO_2-eq/1,000 km. PWB7 is modeled without any combustion of rape straw, which results in an increase of 42 kg CO_2-eq/1,000 km compared to PWB1. PWB8 is modeled without combustion of rape straw but using a longer transportation distance for the rapeseed (from eastern Europe to northern Europe by ship), which results in an increase of 10 kg CO_2-eq/1,000 km compared to PWB7.

It should be noted that from Fig. 2 other combinations are possible to construct than the ones presented. For example, if the gains from increased rape straw combustion in PWB6 are added to PWB2 then the overall impact would decrease even further to ~22 kg CO_2-eq/1,000 km.

Our results are slightly different from the findings by Harding et al. (2008). This difference originates primarily from the rather high climate change potential of chemicals

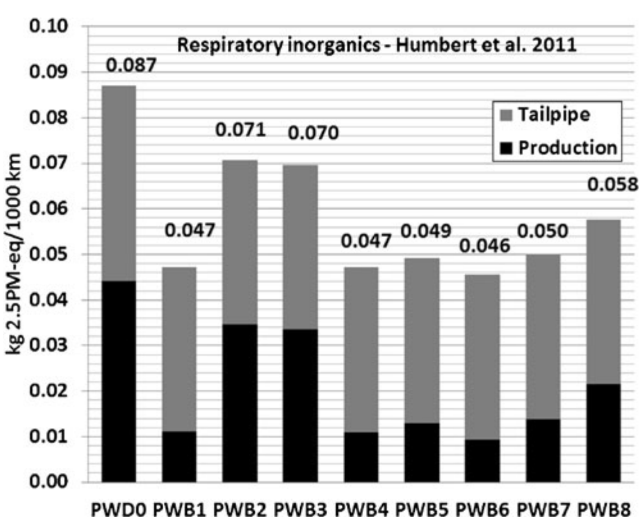

Fig. 4 Respiratory inorganics—Humbert et al. (2011)

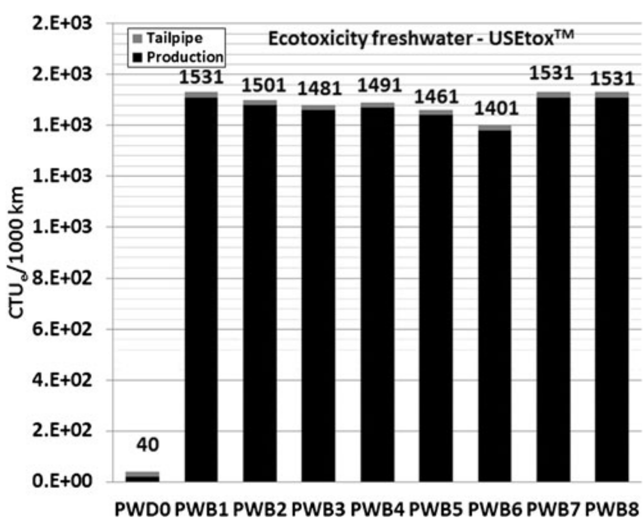

Fig. 6 Ecotoxicity freshwater by USEtox™

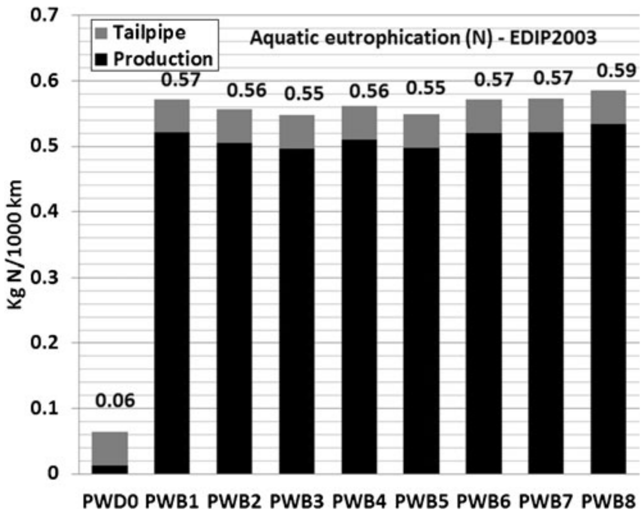

Fig. 7 Aquatic eutrophication (N) using EDIP 2003

used for the conventional transesterification process applied in Harding et al. (2008), compared to the lower climate change potential of chemicals in the conventional transesterification process that we have applied here. However, other differences might also explain the different results between the enzymatic and conventional transesterification processes described in this paper and the results that Harding et al. (2008) present. Furthermore, Harding et al. (2008) arrive at a result ranging from ~147 to 162 kg CO_2-eq/GJ in tank.[2] No combustion of straw is modeled in Harding et al. (2008). PWB1 has climate change potential of ~18 kg CO_2-eq/GJ in tank. The low heat value used in Harding et al. (2008) is 27.1 GJ/t of biodiesel, which seems to be a low estimate compared to the findings by Mehta and Anand (2009) of ~37–38 GJ/t of biodiesel, which is our assumed efficiency as well.

The climate change impact potential for our system seems to be in alignment with Edwards et al. (2008). Edwards et al. (2008) report for FAME an upper quartile value of ~69 kg CO_2-eq/GJ in tank, a medium value of ~49 kg CO_2-eq/GJ in tank, and a lower quartile value of ~25 kg CO_2-eq/GJ in tank. Numbers for FAEE are very similar to the numbers reported for FAME. No combustion of straw is reported, which can explain some of the observed differences. It should be noted that Edwards et al. (2008) address possible increased production, and so those results are not strictly comparable to our study in the sense that when modeling different scopes different results should be expected. For example, Edwards et al. (2008) assume that some of the increased rapeseed production will be placed on lower-quality land than is already established, which then would result in a lower efficiency compared to our study. Bernesson et al. (2004) arrive at similar results ranging from ~30 to 88 kg CO_2-eq/GJ in tank. This range is explained by the different allocation methods, in which the system expansion is in the lower part of this range going from 30 to 35 kg CO_2-eq/GJ in tank. No combustion of straw is reported in Bernesson et al. (2004).

3.2 Land use

The land use impact category has been modeled based on the land occupied directly by the crop needed for driving 1,000 km. For validation of the results, two different impact methodologies have been employed: Recipe and Impact 2002+. It seems that for the different biodiesel PWs, the Recipe estimates are in general ~5 m^2a/1,000 km lower than the estimates from Impact 2002+. We assume that this difference results from differences in the modeling principles of the two impact methodologies. PWD0 differs considerably between the two impact methodologies (by a factor of 2), but for comparison with the biodiesel PWs this problem is negligible—the absolute land use values for PC diesel are very small compared to PWB1 to 8.

Comparison between the different PWs reveals what should be expected. PWB2 and 3 have the highest land use impact, due to land use from crop production for both oil and alcohol (bioethanol). PWB5 has a lower land use impact than PWB1 due to the larger alcohol molecule, which results in slightly better land use efficiency for FAEE compared to FAME. Change in the use of rape straw and transportation, which is reflected in PWB6 to 8, does not change the overall land use impact (Fig. 3).

3.3 Respiratory inorganics potential

The human health impacts from respiratory inorganics have been modeled based on Humbert et al. (2011). In general, the total emissions for the PWBs are lower than for PWD0. For PWD0~ half of the emissions are from tailpipe, while the other half is from production. PWB1 has almost half of the 2.5 PM-eq emission compared to PWD0. Approximately three quarters of the emissions in PWB1 are from tailpipe emissions. Tailpipe emissions are quite similar to each other across the PWBs, and so the differences between the various impacts reflect differences in emissions from the production methods. PWB2 and 3 both have bioethanol as the alcohol input to the transesterification process. The respiratory inorganics environmental impact from production of bioethanol is higher compared to production of methanol, due to traction and transportation, and to production of the fertilizers used in the sugar cane production system. The larger impact from PWB8 compared to PWB1 is explained by the increased transportation of rapeseeds from eastern Europe to northern Europe. The overall impact of using PC methanol (PWB1) instead of PC ethanol (PAW5) seems to be

[2] "In tank" refers to the accumulated impact in a well-to-tank perspective.

marginally better regarding the respiratory impact category. Increasing the use of straw (PWB6) also leads to a slight improvement in the respiratory inorganics impact category compared to PWB1 as a result of reduced coal production (Fig. 4).

3.4 Human toxicity (carc) potential

Human toxicity (carc) is modeled based on the USEtox™ methodology (Rosenbaum et al. 2008; Hauschild et al. 2008). Tailpipe emissions from PWD0 and PWB1 through 8 do not differ much; the major contribution to the human toxicity (HT) impact category is from production of nitrogen, phosphorus, and potassium carbonate—all of which are used for rapeseed cultivation—and from traction in the rapeseed production system. The main change in emissions between the different pathways is in the production systems. In general, production of PC diesel results in less HT than from the biodiesel PWs. Figure 5 shows that the two bioethanol PWs (PWB2 and 3) have a higher HT impact compared to PWB1. PWB3 and 4 indicate that the enzymatic transesterification process is preferable compared to the conventional process (PWB1 and 2). Changing the alcohol from PC methanol (PWB1) to PC ethanol (PWB5) results in a higher HT impact. Increasing the use of rape straw from the rapeseed field (PWB6) can potentially lower the environmental impact compared to (PWB1), which is also confirmed in PWB7 where a slightly higher HT impact is observed due to the change in extraction of coal. From PWB8 it can be observed that additional transportation in the production system increases the impact.

3.5 Ecotoxicity—freshwater potential

Freshwater ecotoxicity is modeled based on the USEtox™ methodology (Rosenbaum et al. 2008; Hauschild et al. 2008). The major difference in the freshwater ecotoxicity impact between PWD0 and PWB1 to 8 is due to the difference in the production systems. Small changes between biodiesel PWB1 to 8 can be observed. PWB6 results in an improvement in ecotoxicity impact compared to PWB1 due to the reduced production of coal. This impact originates almost entirely from the use of pesticides in rapeseed production (Fig. 6).

3.6 Aquatic eutrophication (N) potential

Aquatic eutrophication is modeled based on the EDIP 2003 methodology (Hauschild and Potting 2005). The major difference in the aquatic eutrophication impact observed between PWD0 and PWB1 to 8 results from the difference in production systems. Small changes between the different biodiesel in PWB1 to 8 can be observed. PWB8 has the highest impact due to increased transportation compared to PWB1; this impact results primarily from the rapeseed production system, with additional contribution from the use of fertilizers and traction (Fig. 7).

3.7 Uncertainty consideration

As pointed out by Mathiesen et al. (2009), substitution effects are not certain. In the present assessment, there are uncertainties related to the way system expansion has been carried out in order to solve allocation problems. For example, it has been assumed in our study that glycerol will substitute for wheat. According to Malça and Freire (2011), bio-glycerol can also substitute other products, such as PC glycerol, and in Pagliaro (2010) other products that glycerol can substitute are discussed. Most of these are relevant for future time-periods (prospective), but not for the current time-period addressed here. Based on these references and sensitivity runs in SimaPro, different substitutions can vary the impact potentials from different categories by up to 10–15 %.

Since the presented LCA is comparable to a "still picture" of the present situation, market effects other than the system expansion used to solve the allocation problems are not modeled, such as rebound effects from increased production or increased efficiency. For example, Mulalic (2011) shows that efficiency improvement in truck engines can lead to an overall market increase in fuel consumption due to rebound effects. Another factor that can influence uncertainty when modeling market effects, and hence substitution effects, is whether the market is increasing or decreasing. If the market is increasing, it is plausible that no substitution effect will take place—the product that was assumed to substitute for another product simply becomes an *additional* product on the market. In general, as discussed in Møller (1996), if changes are considered on a macro scale, then changes in the price vector should be considered as well.

Impact from indirect land use change Based on Indirect Land Use Change numbers from Croezen et al. (2010), and energy values for rapeseed oil from Mehta and Anand (2009), impact from ILUC can be added to GHG emissions in PWB1. A "medium ILUC impact" based on these numbers results in an increased emission of ~107 kg CO_2-eq/1,000 km, while a "high ILUC impact" results in an increased emission of ~173 kg CO_2-eq/1,000 km. However, our study addresses only the established production of biodiesel as it is today; it does not address what could happen if the production of biodiesel is increased prospectively. The available ILUC numbers do address what could happen as an indirect effect if rapeseed production is increased. This means that a distinction between these two LCA scopes is important for the interpretation of the results, and for how the results can be used. This concern is also reflected by

Halleux et al. (2008). The results in this present paper should not be interpreted as a blueprint for increased biodiesel production; they represent only options for improvements to already-established production. Furthermore, and as extensively discussed by Gawel and Ludwig (2011), there are several uncertainty issues with the ILUC numbers that would require attention before those numbers can be applied. Two specific issues are causality and how to measure the ILUC impact. Regarding causality, it can be difficult to distinguish between different drivers for land use change. Kline and Dale (2008) list other possible drivers beyond a single-crop market (cultural, technological, biophysical, and economic forces, for example). Measuring or monitoring these different drivers is problematic. For example, the data used in the Global Trade Analysis Project (GTAP)[3] model is based on voluntary reporting, and on data that can be rather old such as Swedish input–output data from 1985 (Reinvang and Peters 2008).

Substitution of coal Given that the substitution of rape straw for coal in a power plant is not a one-to-one ratio measured in gigajoules, this assumption of substitution can affect the results as well. If the substitution effect is zero, then the climate change potential for PWB1 would be 97.8 kg CO_2-eq/1,000 km. Because coal is one of the energy carriers with the highest climate change potential, the substitution effect of any other energy carrier should be expected to be between 97.8 kg CO_2-eq/1,000 km and 57 kg CO_2-eq/1,000 km. However, in Denmark straw is used for co-firing, which seems to be a good reason for assuming that coal is substituted.

CO_2 emission from PWD0 In this paper, emission data has been modeled based on the EcoInvent database. The CO_2-eq emission for PWD0 deviates from other references, possibly due to different assumptions of what has been included or excluded and to inherent uncertainties in these data points. According to the EcoInvent database, 55.8 kg of PC diesel is needed to drive 1,000 km with a standard diesel car. Given that 87 % of PC diesel is carbon (C), then the emission of carbon is:

0.0558 Mg×0.87=0.049 Mg C. 0.049 Mg/(12 Mg/Mmol)~4.05 kmol of C. CO_2=44 g/mol. 4.05 kmol C× 44 kg/kmol CO_2~180 kg CO_2/1,000 km.

According to the EcoInvent database ~34 kg of CO_2-eq is emitted when producing 55.8 kg PC diesel. As such, approximately 85 % of the total emission in PWD0 is from tailpipe emissions—and because this seems to be unavoidable, the uncertainty of this number should be fairly low. The 15–16 % from production of PC diesel might be more uncertain, and might depend on what has been included or excluded in the EcoInvent database compared to other references. In Edwards et al. (2008), 0.0426 kg of PC diesel is estimated to be combusted to drive 1,000 km; this is only 75 % of the amount needed according to the EcoInvent database. Data from the EcoInvent database is assumedly based on other references with other test conditions, such as different types of engines and cars that likely have different characterizations and efficiencies. A more efficient engine would also affect all the PWBs with a similar reduction of fuel consumption, and as a consequence increase the importance of the tailpipe emission element of the total emission.

[3] The GTAP model is used to calculate ILUC impacts according to Hedal et al. (2010).

4 Conclusions

Six different impact categories have been evaluated in a WTW perspective in this study. The main sources of various environmental impacts are summarized here, and options for improvements are suggested.

The recommendations given in this paper are not based on one score aggregating every environmental impact. Rather, each impact category is evaluated separately, and the recommendation is given exclusively for this specific impact category. Environmental impacts are not aggregated into a single score because such a step can potentially exacerbate/introduce uncertainty regarding decision support due to an increased number of embedded assumptions regarding the value of 1 kg of CO_2 versus the value of 1 ha of land. Trade-offs between different impact categories are also not considered.

In PWD0, the primary source of climate change potential originates from tailpipe emissions, with a tailpipe/(production + tailpipe)-ratio of 180/214 kg CO_2-eq/1,000 km (~85 %). The impact from PWD0 is used to benchmark the findings for PWB1 to 8.

Climate change potential For the different biodiesel pathways the main impacts result from the agricultural stage; the use of mineral fertilizer (ammonium nitrate), traction for harvesting, and transport of harvested rapeseed contributes to climate change potential. The potential for significant improvements in this production system comes from increased use of rapes straw for combustion, which is assumed to substitute for coal and lower transportation in the product system. Bioethanol or biomethanol can be used to reduce tailpipe emissions compared to PC ethanol or methanol.

Land use PWD0 represents an insignificant use of land compared to PWB1 to 8. Using bioethanol compared to PC ethanol (or methanol) would increase land use ~15–20 %. If decreasing land use is desired, PC alcohol (and/or oil) is favorable.

Respiratory inorganics potential PWD0 has the largest respiratory inorganics impact potential. Among PWB1 to 8, PWB 2 and 3 have the highest impacts due to the use of bioethanol.

Human toxicity (carc) potential The lowest impact is from PWD0, and among PWB1 to 8 there is some variation. The main sources originate from the production stage for both PC diesel and biodiesel. For PWB1 to 8, the largest contribution results from the use of fertilizer. It is not preferable to change alcohol from PC methanol to bioethanol in terms of human toxicity potential.

Ecotoxicity—freshwater potential PWD0 has the lowest impact, while PWB1 to 8 have more-or-less similar impacts. Almost all of the various impacts come from the production system, in the use of pesticides in rapeseed production.

Aquatic eutrophication (N) potential The major differences in the aquatic eutrophication impact observed between PWD0 and PWB1 to 8 result from differences in the production systems. Small changes among PWB1 to 8 result primarily from the rapeseed production system, from traction and the use of fertilizers.

Based on the analysis in this paper, we recommend investigating the following specific options and incentives.

- From a climate change potential perspective, increasing the use of rape straw for energy production in a power plant, but taking carbon sequestration problems into consideration.
- From a climate change potential perspective, increasing the use of bio-alcohol instead of PC alcohol in the transesterification process.
- From a climate change potential perspective, changing the fuel used in the system from PC fuel to biofuel.
- From a land use perspective, using PC diesel instead of biodiesel.

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References

Bernesson S, Nilsson D, Hansson P (2004) A limited LCA comparing large- and small-scale production of rape methyl ester (RME) under Swedish conditions. Biomass Bioenerg 26(6):545–559

Birkved M, Hauschild M (2006) PestLCI—a model for estimating field emissions of pesticides in agricultural LCA. Ecol Model 198:433–451

Croezen H, Bergsma G, Otten M, Valkengoed M (2010) Biofuels: indirect land use change and climate impact, European Federation for Transport and Environment AISBL, Delft, the Netherlands

Danish Agro (2008) Plant protection 2008 Field journal—book for notes [Planteværn 2008 Markjournal—notatbog i marken]. http://www.danishagro.dk/documents/Plantev%C3%A6rnsbog_web.pdf . Accessed 14 April 2010

Danish Statistics (2011) Rapeseed removel from land fields in Denmark (2006-2009). http://www.statistikbanken.dk/statbank5a/default.asp?w=1280. Accessed 15 February 2011

Edwards R, Larivé J, Mahieu V, Rouveirolles P (2008) Well-to-Wheels Analysis of Future Automotive Fuels and Powertrains in the European Context Well: Tank-to-Wheels Report, European Commission—Joint Research Centre. http://ies.jrc.ec.europa.eu/WTW.html. Accessed 23 February 2010

Emerging Markets Online (2011) Biodiesel 2020: A Global Market Survey. http://www.emerging-markets.com/biodiesel/swf/Europe%20Biodiesel%20Production%20and%20Capacity.html. Accessed 20 April 2011

European Commission (2008) European Energy and Transport—Trends to 2030—Update 2007. http://ec.europa.eu/dgs/energy_transport/figures/trends_2030_update_2007/energy_transport_trends_2030_update_2007_en.pdf. Accessed 28 September 2011

LCA Food Database (2006) Rapeseed meal substitution effects. LCA food database in SimaPro

Gawel E, Ludwig G (2011) The iLUC dilemma: how to deal with indirect land use changes when governing energy crops? Land Use Policy 28(4):846–856

Goedkoop M, Heijungs R, Huijbregts M, Schryver A, Struijs J, Zelm R (2008) ReCiPe 2008 A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Report I. Characterisation factors. Ministry of Housing, Spatial Planning and the Environment, Netherlands

Graboski M, McCormick R, Alleman T, Herring A (2003) The effect of biodiesel composition on engine emissions from a DDC Series 60 diesel engine. National Renewable Energy Laboratory, Golden

Haastrup M (2010) Winter rapeseeds, cultivation guide [Vinterraps, dyrkningsvejledning], Videncentret for Landbrug, Planteproduktion, Aarhus Universitet

Halleux H, Lassaux S, Renzoni R, Germain A (2008) Comparative life cycle assessment of two biofuels ethanol from sugar beet and rapeseed methyl ester. Int J Life Cycle Assess 13(3):184–190

Harding K, Dennis J, Blottnitz H, Harrison S (2008) A life-cycle comparison between inorganic and biological catalysis for the production of biodiesel. J Clean Prod 16(13):1368–1378

Hauschild M, Potting J (2005) Spatial differentiation in life cycle impact assessment—the EDIP 2003 methodology. Danish Ministry of the Environment, Environmental Protection Agency Environmental News no. 80, Copenhagen

Hauschild M, Huijbregts M, Jolliet O, MacLeod M, Margni M, Rosenbaum R, McKone T (2008) Building a model based on scientific consensus for life cycle impact assessment of chemicals: the search for harmony and parsimony. Environ Sci Technol 42(19):7032–7037

Hedal J, Baltzer K, Nielsen P (2010) Life cycle inventory modelling of land use induced by crop consumption. Int J Life Cycle Assess 15 (1):90–103

Howarth R, Bringezu S (2009) Biofuels. International SCOPE Biofuels Project, United Nations Foundation, Deutsche Forschungsgemeinschaft, David and Lucile Packard Foundation, United Nations Environment Programme, Cornell Center for a Sustainable Future, Biogeochemistry & Biocomplexity Initiative at Cornell University & Wuppertal Institut für Klima, Umwelt und Energie, Cornell University, Ithaca, NY

Humbert S, Marshall J, Shaked S, Spadaro J, Nishioka Y, Preiss P, McKone T, Horvath A, Jolliet O (2011) Intake fraction for particulate matter: recommendations for life cycle impact assessment. Environ Sci Technol 45(11):4808–4816

International Energy Agency (2005) Energy statistics manual. International Energy Agency. http://www.iea.org/publications/free_new_Desc.asp?PUBS_ID=2367. Accessed 28 November 2011

ISO (2006a) ISO 14040 Environmental managements—life cycle assessments—principles and framework. International Organisation for Standardisation, Geneva

ISO (2006b) ISO 14044 Environmental managements—life cycle assessments—requirements and guidelines. International Organisation for Standardisation, Geneva

Jensen K, Thyø A, Wenzel H (2007) 2nd generation bioethanol for transport: the IBUS concept—boundary conditions and environmental assessment. Technical University of Denmark

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, Rosenbaum R (2003) IMPACT 2002+: a new life cycle impact assessment methodology. Int J Life Cycle Assess 8(6):324–330

Jonasson K, Sandén B (2004) Time and scale aspects in life cycle assessment of emerging technologies—case study on alternative transport fuels. CPM—Centre for Environmental Assessment of Product and Material Systems. Chalmers University of Technology, Göteborg

Kline K, Dale H (2008) Biofuels: effects on land and fire. Science (Washington DC) 321(5886):199–201

Lafond G, May W, Stumborg M, Lemke R, Holzapfel C, Campbell C (2009) Quantifying straw removal through baling and measuring the long-term impact on soil quality and wheat production. Agron J 101(3):529–537

Makridakis S, Wheelwright S, Hyndman R (1998) Forecasting: methods and applications, 3rd edn. Wiley, New York

Malça J, Freire F (2011) Life-cycle studies of biodiesel in Europe: a review addressing the variability of results and modeling issues. Renew Sust Energ Rev 15(1):338–351

Mathiesen B, Münster M, Fruergaard T (2009) Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments. J Clean Prod 17(15):1331–1338

Mattson F, Volpenhein R (1963) Specific distribution of unsaturated fatty acids in triglycerides of plants. J Lipid Res 4(4):392–396

Mehta P, Anand K (2009) Estimation of a lower heating value of vegetable oil and biodiesel fuel. Energ Fuel 23(8):3893–3898

Møller F (1996) Valuation of environmental goods [Værdisætning af miljøgoder]. Jurist- og Økonomforbundets Forlag, København

Møller S, Børsting C, Hansen N, Lassén M, Sandbøl P, Schack P (2001) Nutrient excretion from fur animals—nitrogen, phosphorus and potassium in manure—basic numbers 2000 [Næringsstofudskillelse fra pelsdyr—kvælstof, fosfor og kalium i husdyrgødning—normtal 2000]. DJF Rapport nr. 36, Husdyrbrug, Aarhus Universitet, Det Jordbrugsvidenskabelige Fakultet

Montgomery D (2005) Design and analysis of experiments, 6th edn. Wiley, Hoboken

Mulalic I (2011) The determinants of fuel use in the trucking industry—volume, size and the rebound effect. 6th Kuhmo Nectar Conference and Summer School on Transportation Economics

Munkholm L, Sibbesen E (1997) Theme: loss of phosphorus from agricultural land [Tema: tab af fosfor fra landbrugsjord]. Det strategiske Miljøforskningsprogram, Århus

Naturerhvervsstyrelsen (2010) Guidance on fertilization and rules for harmonization—period: August 9, 2009—July 31, 2010 [Vejledning om gødsknings-og harmoniregler—Planperioden 1. August 2009 til 31. juli 2010]. Vejledning edn, Ministeriet for Fødevarer, Landbrug og Fiskeri, Naturerhvervsstyrelsen, Denmark

Nielsen P, Oxenboll K, Wenzel H (2007) Cradle-to-gate environmental assessment of enzyme products produced industrially in Denmark by Novozymes A/S. Int J Life Cycle Assess 12(6):432–438

Pagliaro M, Royal Society of Chemistry (Great Britain) (2010) The future of glycerol, 2nd edn. Royal Society of Chemistry, Cambridge

Reinvang R, Peters G (2008) Norwegian consumption, Chinese pollution. An example of how OECD imports generate CO_2 emissions in developing countries. Norwegian University of Science and Technology, WWF Norway, WWF China Programme Office

Rosenbaum R, Bachmann T, Gold L, Huijbregts M, Jolliet O, Juraske R, Koehler A, Larsen H, MacLeod M, Margni M, McKone T, Payet J, Schuhmacher M, Meent D, Hauschild M (2008) USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess 13(7):532–546

Sotoft L, Rong B, Christensen K, Norddahl B (2010) Process simulation and economical evaluation of enzymatic biodiesel production plant. Bioresource Technol 101(14):5266–5274

The European Parliament and the Council (2009) Promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC, Directive edn, Official Journal of the European Union

Vinther F, Hansen (2004) SimDen: a simple model for quantification of N_2O-emissions and denitrification [SimDen: en simpel model til kvantificering af N_2O-emission og denitrifikation]. Ministeriet for Fødevarer, Landbrug og Fiskeri, Danmarks JordbrugsForskning, Forskningscenter Foulum, Tjele, Denmark

Zijlstra R, Menjivar K, Lawrence E, Beltranena E (2009) The effect of feeding crude glycerol on growth performance and nutrient digestibility in weaned pigs. Can J Anim Sci 89(1):85–89